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Article

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Global Change Pressures on Soils from Land Use and Management

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Abstract

Soils are subject to varying degrees of direct or indirect human disturbance, constituting a major global change driver. Factoring out natural from direct and indirect human influence is not always straightforward, but some human activities have clear impacts. These include land use change, land management, and land degradation (erosion, compaction, sealing and salinization). The intensity of land use also exerts a great impact on soils, and soils are also subject to indirect impacts arising from human activity, such as acid deposition (sulphur and nitrogen) and heavy metal pollution. In this critical review, we report the state-of-the-art understanding of these global change pressures on soils, identify knowledge gaps and research challenges, and highlight actions and policies to minimise adverse environmental impacts arising from these global change drivers.

Soils are central to considerations of what constitutes sustainable intensification. Therefore, ensuring that vulnerable and high environmental value soils are considered when protecting important habitats and ecosystems, will help to reduce the pressure on land from global change drivers. To ensure that soils are protected as part of wider environmental efforts, a global soil resilience programme should be considered, to monitor, recover or sustain soil fertility and function, and to enhance the ecosystem services provided by soils. Soils cannot, and should not, be considered in isolation of the ecosystems that they underpin and vice versa. The role of soils in supporting ecosystems and natural capital needs greater recognition. The lasting legacy of the International Year of Soils in 2015 should be to put soils at the centre of policy supporting environmental protection and sustainable development.

1. Introduction

2015 is the International Year of Soil. This represents an ideal time to take stock of scientific knowledge about the changing global pressures that humans are exerting on soils. 2015 is also the year when policy makers will adopt a new legally-binding climate agreement under the United Nations Framework Convention on Climate Change (UNFCCC), with individual countries and businesses making policies and targets on greenhouse gas emissions and removals. Soils storage and cycling of carbon and nitrogen are part of emissions and removals from the land sector. Furthermore, 2015 is the year when countries will shape and adopt a new development agenda that will build on the Millennium Development Goals (MDGs). With increasing population, issues such as food security, water security, energy security (including bioenergy production) and sustainable integrated land and resource management are central to many development research and policy agendas. Soils underpin the provision of many ecosystem services related to development.

Soils provide multiple ecosystem services, allowing sustained food and fibre production, and delivering climate regulation, flood regulation, improved air and water quality, reducing soil erosion, and provide a reservoir for biodiversity (Smith et al. 2015). All soils are subject to some degree of human disturbance, either directly through land-use and land management, or indirectly through responses to human-induced global change such as pollution and climate change. Distinguishing natural from direct and indirect human influence is not always straightforward (Smith, 2005), but some human activities and their consequences have clear impacts, and despite large heterogeneity in soil properties and responses, robust scientific knowledge exists.

Human impacts on soils largely emerge from the need to meet the food, fibre, and fuel demands of a growing population including an increase in meat consumption as developing nations become wealthier, the production of biofuels, and increasing areas of urbanization. This has led to conversion of natural land to managed land (extensification) and intensification of agricultural and other management practices on existing land such as increasing nutrient and water inputs and increasing harvest frequency to increase yields per hectare.

Land cover or land use change (e.g. from forest or natural grassland to pasture or cropland), removes biomass, changes vegetation and disturbs soils, leading to loss of soil carbon and other nutrients, changes in soil properties, and changes to above- and below-ground biodiversity. Some land cover conversions e.g. reforestation after abandonment of cropland, can increase both above- and below-ground carbon and nutrients. Land use or land management that does not result in a change of cover (e.g. forest harvest and regrowth, increased grazing intensity and intensification of crop production), can potentially result in degradation of soil properties, depending on the characteristics of the management practices.

Land use change has been accelerated by population increases and migration as food, shelter, and materials are sought and acquired. It is estimated that humans have directly modified at least 70 Mkm^2, or >50 percent of Earth's ice-free land area (Hooke et al. 2012). The new Global Land Cover Share-database (Latham et al., 2014) represents the major land cover classes defined by the FAO. Croplands and grasslands (including both natural grasslands and managed grazing lands) each covered 13.0 %. "Tree-covered areas" (i.e. both natural and managed forests) covered 28%, shrub-covered areas 9.5 %. Artificial surfaces (including urbanised areas) occupy 1 %. Land degradation can be found in all land cover types. Degraded land covers approximately 24% of the global land area (35 Mkm^2). 23% of degrading land is under broadleaved forest, 19% under needle-leaved forests and 20-25% on rangeland (Bai *et al*., 2008).

In this review we report the state-of-the-art understanding, and knowledge gaps concerning impacts of changes in anthropogenic land use and land management on soils, including interactions with other anthropogenic global change pressures. We also review actions and policies that limit the adverse impacts arising from these global change drivers. We make the case to put soils at the centre of research strategy and policy actions as a legacy of the International Year of Soils.

2. Land use/land cover change

Land cover change has been dominated by deforestation, but also conversion of grasslands to cropland and grazing land. Deforestation has had the greatest impact on historical soil carbon change, causing on average around 25% of soil carbon to be lost (Guo & Gifford, 2002; Murty *et al*., 2002). Soil carbon losses largely stem from oxidation of the organic matter as well as soil erosion.

Deforestation affected an estimated 13 million hectares per year between 2000 and 2010; net forest loss was 5.2 million hectares per year (FAO, 2010). Most of this recent deforestation has taken place in tropical countries (FAO, 2010; Hansen et al., 2013). Over 50% of tropical forest loss occurred in Brazil and Indonesia, largely driven by a few commodities: timber, soy, beef, and oil palm (West *et al*., 2014). There has been a reduced rate of deforestation in some regions over the last decade, most notably Brazil (INPE, 2014), largely because of land use conservation policies (Soares-Filho *et al*., 2014; Nolte *et al*., 2013) as well as economics. Most developed countries with temperate and boreal forest ecosystems – and more recently, countries in the Near East and Asia – are experiencing stable or increasing forest areas in contrast to the large scale historic deforestation in these regions, with afforestation reported in Europe, USA, China, Vietnam and India (FAO 2013).

Changes in soil properties can vary markedly with type of land cover change, climate, and method, extent of vegetation removal (e.g. land clearing, fires, mechanical harvest) and management post-harvest. For example, West *et al*. (2010) estimated that clearing land in the tropics generally emits three times the amount of carbon per ton of annual crop production compared to clearing land in temperate areas. Emissions are particularly high when organic peatland/wetland soils are drained to enable agriculture as the initial soil carbon is higher, and drainage results in large losses of carbon as previously anaerobic soils become aerobic, allowing the organic matter to oxidise. For example, clearing forest on organic soils for palm oil production in Kalimantan emits nine-times more carbon than clearing on neighbouring mineral soils (Carlson & Curran, 2013). Impacts of deforestation can be reduced by avoiding deforestation on organic soils, and on steep slopes prone to erosion.

There is large heterogeneity in soil measurements of carbon, nitrogen, microbes etc., and still many areas of the world with poor data coverage. Models can be used to fill gaps in spatial coverage and look at past and future time periods, but these too give very variable results. Nevertheless there are some clear signals that can be obtained from meta-analyses of field data and global model results of land use/land cover change with respect to soil carbon.

2.1. Observations of impacts of land cover change

Table 1 presents the results of different meta-analysis studies across different climatic zones that compared the impacts of land use changes on SOC (Guo & Gifford 2002; Don *et al.* 2011; Poeplau *et al.* 2011; Bárcena *et al.* 2014; Murty *et al.* 2002; Wei *et al.* 2014a). Changes in SOC after the conversion of forests to croplands ranged from -24 to -52% without marked differences between climatic regions. The conversion of pastures to other uses (tree plantations and particularly, croplands) also induced decreases in SOC (-10% and -59%, respectively). On the other hand, the substitution of croplands by other land uses (forest regrowth, tree plantation, grassland, pasture) resulted in an increase of SOC (+18 to +53%). In the case of afforestation, soil C increase with time after afforestation, and C sequestration depends on prior land use, climate and the tree species planted.

Fewer meta-analysis studies are available for changes in soil N with changes in land uses. A compilation with predominance of data from tropical sites indicated that average loss of 15% of soil N after conversion of forests to croplands (Murty *et al.* 2002). In Australia, N losses after conversion of native vegetation to perennial pasture and cropland were more than 20% and 38%, respectively (Dalal *et al.* 2013) while in China N loss (0-10 cm depth) was 21% and 31% after 4 and 50 years after conversion of forests to cropland (Wei *et al.* 2014b). Similarly to what was described for SOC, afforestation in subtropical zone results in a significant increase of N stocks 50 years after conversion (Li *et al.* 2012).

[Table 1 here]

2.2. Modelled impacts of land cover change

Dynamic Global Vegetation Models (DGVMs) are used to look at the combined effects of land use change, climate, CO_2, and in some cases N deposition, on vegetation and soil properties over time. A few global models include some aspects of forest, grassland or cropland management (Bondeau *et al.* 2007; Lindeskog *et al.* 2013; Drewniak *et al.* 2013; Jain *et al.* 2005). Most DGVMs do not currently model peatland soils. In Tables 1 and 2, and Figures 1 and 2, we show impacts of past land cover and management change on soil carbon and nitrogen as calculated by three DGVMs: ISAM (Jain *et al.* 2013; El-Masri *et al.* 2013; Barman *et al.* 2014 a,b); LPJ-GUESS (Smith *et al.* 2001; Lindeskog *et al.* 2013); and LPJmL (Bondeau *et al.* 2007). The ISAM and LPJ-GUESS models were run with the HYDE historical land use change data set (History Database of the Global Environment; Klein

Goldewijk *et al.* 2011). ISAM included wood harvest following (Hurtt *et al.* 2011). The LPJmL group combined 3 land use change data sets with the geographic distribution of global agricultural lands in the year 2000. All models were run with historical climate and CO_2, and additionally N deposition in the ISAM model only as it includes a nitrogen cycle. The effects of land cover change were isolated by comparing model runs with and without land use/management (Le Quéré *et al.* 2014). Table 2 and Figure 1 show the loss of soil carbon due to historical land use change from 1860 to 2010 (note there was land use change causing soil carbon loss prior to 1860 particularly in Europe and central Asia, but results are not shown as they were not available for all three models). As with the observed data (Table 1) high carbon losses are associated with the conversion of forests to croplands. Figure 2 shows the mineral soil C and N concentration of different land cover types in different geographic ranges.

[Figure 1 & 2; Table 2 here]

Differences between the models are large for some systems and regions due to different land use change data, different land cover definitions, and different processes included in the models. For example, soil carbon losses are higher in the LPJmL model (Table 2, Figure 1) in part due to greater land cover change in their land cover reconstructions, while their boreal grassland soil carbon is high due to the inclusion of permafrost slowing soil carbon decomposition (Figure 2). Treatment of management processes turns out to be an important differentiator. ISAM shows strong decreases of soil carbon in some regions e.g. the southern Boreal zone (Figure 1) where the inclusion of wood harvest removes carbon and nutrients from the soil, while increases in soil carbon in parts of the mid.-latitudes are due to regrowth of forest following abandonment of agricultural land.

In semi-arid to arid regions, LPJ-GUESS and LPJmL show opposite signs of soil carbon change after conversion of natural land to pastures (Figure 1), primarily because LPJ-GUESS simulates a greater fraction of woody vegetation than LPJmL in these regions under potential natural vegetation. Conversion of woody vegetation to pasture slightly increases soil carbon (see the meta analysis of Guo & Gifford 2002), partly because of boosted productivity and higher turnover rates adding more C to the soil, while the change from potential natural grassland to managed pasture (for which the literature is sparse) results in a soil carbon decrease in LPJmL Pasture management strategies can have a large influence on the soil carbon storage (see Section 4.3), and may also be partly be responsible for differences.

Vegetation models are embedded in Earth System Models (ESMs) used to project future climates under different human activity including different land management. Some significant differences between future model climate projections stem from the differences in modeling soil carbon, in particular, the strength of the relationship between increasing temperatures and the increasing rate of soil carbon decomposition (Q_{10}) causing climate-carbon feedbacks *via* CO_2 emissions (Friedlingstein *et al.* 2006). A recent intercomparison of 11 ESMs used in the IPCC 5th Assessment Report (Todd-Brown *et al.* 2013), found the estimate of global soil carbon from ESMs ranged from 510 to 3040 PgC across 11 ESMs compared to an estimate of 890-1600 PgC (95% confidence interval) from the Harmonized World Soil Data Base (FAO/IIASA/ISRIC/ISSCAS/JRC, 2012), with all models having difficulty representing the spatial variability of soil carbon at smaller (1 degree) scales compared to empirical data. In all models NPP and temperature strongly influenced soil carbon stocks, much more so than in the observational data, and differences between models was found to be largely due to the representation of NPP and the parameterization of soil decomposition sub-models. A similar, systematic analysis of DGVMs including benchmarking with observational data, and careful testing of assumptions and process representations in these models, making use of the very large number of observations that have become available in the years since these algorithms were formulated (e.g. Medlyn *et al.* 2015), could significantly improve model performance. This, along with better representation of critical biological and geochemical mechanisms would improve model capability (Todd-Brown *et al.* 2013).

2.3 Drainage and conversion of peatlands/wetlands for agriculture

The organic soils in peatlands/wetlands store vast quantities of carbon which decomposes rapidly when they are drained for agriculture or commercial forestry, resulting in emissions of CO_2 and N_2O to the atmosphere (Hooijer *et al*., 2010). Other services, in particular water storage and biodiversity, are negatively impacted. Drainage increases vulnerability to further losses through fire. The majority of soil carbon is concentrated in peatlands in the boreal zone and tropical peatland forests in Southeast Asia. These areas, along with wetlands along the banks of rivers, lakes and estuaries have increasingly been developed for croplands/bioenergy production over recent decades. The FAO emissions database estimates that globally there are 250 000 km^2 of drained organic soils under cropland and grassland, with total GHG emissions (N_2O plus CO_2) of 0.9 Pg CO_2eq yr^{-1} in 2010, with the largest contributions from Asia (0.44 Pg CO_2eq yr^{-1}) and Europe (0.18 Pg CO_2eq yr^{-1}; FAOSTAT, 2013; Tubiello *et al.*, 2015). Joosten (2010) estimated that there are >500 000 km^2 of drained peatlands in the world, including under forests, with CO_2 emissions having increased from 1.06 Pg CO_2 yr^{-1} in 1990 to 1.30 Pg CO_2 yr^{-1} in 2008, despite a decreasing trend in developed countries, from 0.65 to 0.49 Pg CO_2 yr^{-1}, primarily due to natural and artificial rewetting of peatlands. In Southeast Asia, CO_2 emissions from drained peatlands in 2006 were 0.61 ± 0.25 Pg CO_2 yr^{-1} (Hooijer *et al.*, 2010). Conversion of peatlands in Southeast Asia is increasing, particularly for oil palm expansion, where cleared peatlands typically emit ~9 times more carbon than neighbouring mineral soils (Carlson & Curran 2013). In China, between 1950 and 2000, 13 000 km^2 of wetland soils were shifted to cultivated arable lands, which led to a SOC loss of 5.5 Pg CO_2, mostly from peatlands in Northeast China and Tibet (Zhang *et al.*, 2008).

Soil drainage also affects mineral soils. Meersmans *et al*. (2009) showed that initially poorly drained valley soils in Belgium have lost significant amount of topsoil SOC (i.e. between ~70 and 150 t CO_2 ha^{-1} over the 1960 – 2006 period), most probably as a consequence of intensified soil drainage practices for cultivation purposes.

3. Agricultural management

To meet projected increases in food demand, crop production will need to increase by 70-110% by 2050 (World Bank, 2008; Royal Society of London, 2009; Tilman *et al.*, 2011). This can be achieved either through further expansion of agricultural land (extensification), or through intensification of production on existing land. Intensification is widely promoted as the more sustainable option because of the negative environmental consequences of land expansion through deforestation and wetland cultivation (Foley *et al.*, 2011). For example, Burney *et al*. (2010) estimate that intensification of production on croplands between 1961 and 2010 avoided the release of 590 PgCO_2eq. Increased productivity per unit land area can be achieved through a variety of management practices, such as fertilization, irrigation and increased livestock density, but these can lead to adverse consequences for the soil and wider environment (Tilman *et al.*, 2002). Intensifying land use can potentially reduce soil fertility (without additional inputs) and its ability to sustain high production, as well as soil resilience to extreme weather under climate change, pests and biological invasion, environmental pollutants and other pressures. Some key management practices and consequences are highlighted below and summarised in Table 3.

[Table 3 here]

3.1 Nutrient management

Cultivation of soils results in a decline in soil nutrients (nutrient mining). Nutrient inputs, from both natural and synthetic sources, are needed to sustain soil fertility and supply nutrient requirements for crop production. Nutrient supply can improve plant growth which increases organic matter returns to the soil, which in turn can improve soil quality (see section 3.5), so balanced nutrient supply has a positive impact on soils (Smith et al., 2015). Overuse, however, has negative environmental consequences. Annual global flows of nitrogen and phosphorus are now more than double natural levels (Matson *et al*., 1997; Smil, 2000; Tilman *et al*., 2002). In China, for example, N input in agriculture in the 2000's was twice that in 1980's (State Bureau of Statistics-China, 2005).

Between 50-60% of nutrient inputs remain in agricultural soils after harvest (West *et al*., 2014) and can enter local, regional, and coastal waters becoming a major source of pollution such as eutrophication leading to algal blooms (Carpenter *et al*., 1998). In many places around the world, over-use of synthetic nitrogen fertilizers is causing soil acidification and increased decomposition of soil organic matter, leading to loss of soil function in over-fertilized soils (Ju *et al*., 2009; Tian *et al*., 2012).

Use of fertilisers and manures contributes to climate change through their energy intensive production and inefficient use (Tubiello et al., 2015). Globally, approximately 3-5% of nitrogen additions are released as nitrous oxide (N_2O) to the atmosphere when both direct (from soils) and indirect (e.g. downstream from nitrate leaching) emissions are considered (Galloway *et al*., 2004), and N_2O has ~300 times the radiative forcing of carbon dioxide (IPCC, 2007). Recent research indicates that the relationship between nitrogen application and N_2O emissions is non-linear, resulting in an increasing proportion of added N being emitted, as application rate increases (Philibert *et al*., 2013; Shcherbak *et al*., 2014). China, India, and the United States account for ~56% of all N_2O emissions from croplands, with 28% from China alone (West *et al*., 2014). Overuse of nitrogen and phosphorus fertilizer can contribute to eutrophication of water bodies, adversely affecting water quality and biodiversity (Galloway *et al.*, 2003, 2004, 2008).

Nutrient use-efficiency can be significantly increased, and nitrate losses to water and N_2O emissions can be reduced, through changes in rate, timing, placement, and type of application, as well as balancing fertilization (Venterea *et al*., 2012; Snyder *et al*., 2014). It has been estimated that current levels of global cereal production could be maintained while decreasing global nitrogen application by 50% (Mueller *et al*., 2014).

3.2 Carbon management: reduced disturbance and organic matter additions

Agricultural soils have the potential to store additional carbon than at present if best management practices are used (Paustian *et al.*, 1997; Smith, 2008; Smith, 2012). Soil organic matter content of soils can be increased through use of improved crop varieties or grassland species mixtures with greater root mass or deeper roots (Kell, 2012), improved crop rotations in which C inputs are increased over a rotation (Burney *et al.*, 2010), greater residue retention (Wilhelm *et al.*, 2004), and use of cover crops during fallow periods to provide year-round C inputs (Burney *et al.*, 2010; Poeplau & Don, 2015). Several studies report that soil carbon increases in croplands under no-till management (West & Post, 2002; Ogle *et al*., 2005). However, the carbon benefits of no-till may be limited to the top 30cm of soil (Blanco-Canqui & Lal 2008; Powlson *et al*., 2014). Baker *et al*. (2007) found that total soil carbon was similar in non-till and conventional systems, suggesting that carbon accumulation is occurring at different depths in the soil profile under different management schemes. Given the larger variability in sub-surface horizons and lack of statistical power in most studies, more research is needed on soil carbon accumulation at depth under different tillage regimes (Kravchenko & Robertson, 2010).

Adding plant-derived carbon from external sources such as composts and biochar can increase soil carbon stocks. Composts and biochars are more slowly decomposed compared to fresh plant residues, with mean residence times several (composts) to 10-100 (biochars) longer than un-composted organic materials (Ryals *et al.*, 2015; Lehmann *et al.*, 2015). Recent developments suggest that biochar, from the pyrolysis of crop residues or other biomass, can consistently increase crop N use efficiency while greatly (over 25%) reducing direct N_2O emissions from N fertilizers (Liu *et al.*, 2012; Huang *et al.*, 2012), as well as enhancing soil fertility (Woolf *et al*., 2010).

3.3 Water management

The amount of irrigated croplands has doubled in the last 50 years and now accounts for 70% of all water use on the planet (Gleick, 2003). While irrigated crops cover 24% of all cropland area, they account for 34% of all production (Siebert & Döll, 2010). Irrigation is concentrated in precipitation-limited areas such as India, China, Pakistan, and the USA, which account for 72% of irrigation water use (West *et al*., 2014). Agricultural water-use competes with uses for human and natural ecosystems exacerbating water stress in dry regions. Increased irrigation has occurred in many areas of world agriculture due to the increasing frequency of drought under the climate change (West *et al.*, 2014). Where irrigation increases productivity (e.g. in drought prone areas), organic carbon inputs to the soils would be expected to increase, increasing soil organic matter content (section 3.2).

Irrigation can increase soil salinity in dry regions with high salt content in the subsoil (Ghassemi *et al*., 1995; Setia *et al*., 2011). Where salinization occurs, additional irrigation is needed to "flush" the salts beyond the root zone of the crops, which can further exacerbate stress on water resources, particularly when using underground water sources. Saline soils, which have a high concentration of soluble salts, occupy approximately 3.1% (397 Mha) of the world's land area (FAO, 1995). Climate change (need for more frequent irrigation) and increases in human population (increasing demand for more production) are likely to increase the extent of saline soils (Rengasamy, 2008). The energy required by plants or soil organisms to withdraw water from the soil or retain it in cells increases with decreasing osmotic potential. As soils dry out, the salt concentration in the soil solution increases (decreasing osmotic potential), so two soils of different texture may have the same electrical conductivity, but the osmotic potential is lower in the soil with low water content (Setia *et al.*, 2011a; Chowdhury *et al.*, 2011; Ben-Gal *et al.*, 2009). The accumulation of salts in the root zone has adverse effects on plant growth activity, not only due to negative osmotic potential of the soil solution resulting in decreased availability of water to plants, but also ion imbalance and specific ion toxicity (Chowdhury *et al.*, 2011). Salinity affects microorganisms mainly by decreasing osmotic potential, which affects a wide variety of metabolic activities and alters the composition and activity of the microbial community (Chowdhury *et al.*, 2011) and thereby soil organic matter decomposition.

In saline soils, SOC content is influenced by two opposing factors: reduced plant inputs which may decrease SOC, and reduced rates of decomposition (and associated mineralisation of organic C to CO_2) which could increase SOC content if the C input were unchanged. Using a modified Rothamsted Carbon model (RothC) with a newly-introduced salinity decomposition rate modifier and a plant input modifier (Setia *et al.*, 2011b, 2012), Setia *et al.* (2013) estimated that, historically, world soils that are currently saline have lost an average of 3.47 t SOC ha^{-1} since they became saline. With the extent of saline soils predicted to increase under the future climate, Setia et al. (2013) estimated that world soils may lose 6.8 Pg SOC due to salinity by the year 2100. Soil salinization is difficult to reverse, but salt tolerant plant species could be used to rehabilitate salt affected soils (Setia et al., 2013).

Water efficiency can be improved through management practices that reduce water requirement and evaporation from the soil (such as adding mulch as groundcover), more precise irrigation scheduling and rates, fixing leaks in dryland irrigation systems, improved application technology (e.g., drip irrigation) and use of intermittent irrigation in rice paddies. Given that water limitation is projected to become even more limiting in several semi-arid regions, e.g. Sub-Saharan Africa, where the human population will probably increase most in the future, and climate change impacts are projected to be severe, improved water harvesting methods, e.g. storage systems, terracing and other methods for collecting and storing runoff, are required to make best use of the limited water resource.

3.4 Harvest frequency

Approximately 9% of crop production increases from 1961-2007 was from increasing the harvest frequency (Alexandratos & Bruinsma, 2012). The global harvested area (i.e. counting each time an area is harvested) increased four times faster than total cropland area between 2000 and 2011 (Ray & Foley, 2013). The fraction of net primary production (NPP) extracted by humans is increasing (Haberl *et al.*, 2007). Global warming is increasing the total area suitable for double or even triple cropping in subtropical and warm temperate regions (Liu *et al.*, 2013). The increase results from fewer crop failures, fewer fallow years, and an increase in multi-cropping.

Increasing harvest frequency can reduce soil quality by e.g. continuously removing soil nutrients and increasing soil compaction through greater soil traffic, but if legumes are included in rotations as harvest frequency increases, soil quality could be improved. Increasing harvest frequency may require increasing pesticide and herbicide use, and increased use of fertilisers contributing to pollution (section 3.1). The net effect will depend on the effectiveness of the management practices followed.

3.5 Soil compaction

Soil compaction causes degradation of soil structure by increasing soil bulk density or decreasing porosity through externally or internally applied loads, as air is displaced from the pores between the soil grains (McCarthy, 2007; Alakukku, 2012). It is the most important subtype of physical soil deterioration, covering 68 Mha globally when first mapped in the 1990s (Oldeman et al., 1991). Compaction of agricultural soils often results from heavy machinery or from animal trampling, so is more likely to occur in intensive agricultural systems (machinery use and high stocking densities), and affects physical, chemical and biological properties of soil. Top soil compaction can be reversed and controlled, but when compaction creates impermeable layers in the subsoil, this is less easily reversed.

Subsoil compaction can disrupt nutrient water flows, which in turn can lead to reduced crop yields, poorer crop quality and can give rise to increased GHG emissions, water and nutrient run-off, erosion, reduced biodiversity and reduced groundwater recharge (Batey, 2009). Where compaction cannot be avoided, mitigation is necessary. Biological approaches to mitigation include planting deep rooted plants such as agroforestry; chemical methods include fertilization (to overcome yield penalty, though not to remedy compaction); and technical measures include machinery in which planting does not coincide with wheel tracks, wide tyres / reduced tyre pressures to reduce pressure per unit area, and precision farming to retain the same wheel tracks each year (Hamza & Anderson, 2005).

3.6 Livestock density

Livestock production is projected to increase significantly in order to meet the growing demand from a growing population and increase in per-capita meat consumption, with total demand for meat expected to grow by more than 200 Mt by 2050 (Alexandratos & Bruinsma, 2012). The greatest increases in per-capita consumption are projected to be in developing and transition countries (Alexandratos & Bruinsma, 2012). Since the 1970s, most increased livestock production has resulted from intensification: increasing livestock density and shifting to a greater fraction of livestock raised in industrial conditions (Bouwmann *et al*., 2006). For example, 76-79% of pork and poultry production is industrialized (Herrero & Thornton, 2013). Manure, inputs for growing feed, and soil loss from intensively managed areas can be major sources of water pollution to local and downstream freshwater ecosystems. Clearing natural ecosystems for new pastures, particularly in arid and semi-arid regions, typically occurs on low-productivity lands with a much higher risk of soil erosion and soil carbon/nutrient depletion (Alexandratos & Bruinsma, 2012), and negatively impacts water storage and biodiversity. The impacts of livestock production are particularly prevalent for beef production, which has a least an order of magnitude greater impact on land, water, GHGs, and reactive nitrogen compared to other livestock (Eshel *et al*., 2014; Ripple *et al*., 2014). Moreover, industrial livestock production had led to an increased use of veterinary medicines, antibiotics and hormones, posing potential risks to soil, water, ecosystems and human health. Improved grazing management (e.g. optimised stocking density) can reduce soil degradation, and thereby maintain and enhance organic matter content (McSherry & Ritchie, 2013; see sections 3.2 and 4.3), and can reduce soil compaction, thereby increasing infiltration and water storage and reduce risk of runoff and flooding downstream (Marshall *et al*., 2009).

4. Other land management

4.1 Forest management

Logging and fire are the major causes of forest degradation in the tropics (Bryan *et al*., 2013). Logging removes nutrients and negatively affects soil physical properties and nutrient levels (soil and litter) in tropical (e.g. Olander *et al*., 2005; Villela *et al*., 2006; Alexander, 2012) and temperate forests (Perez *et al*., 2009). Forest Fires affect many physical, chemical, mineralogical, and biological soil properties, depending on fire regime (Certini, 2005). Increased frequency of fires contributes to degradation, and reduces the resilience of the biomes to natural disturbances. A meta-analysis of 57 publications (Nave *et al*., 2011) showed that fire caused a significant decrease in soil C (-26%) and N (-22%). Fires reduced forest floor storage (pool sizes only) by an average of 59% (C) and 50% (N), but the relative concentrations of these two elements did not change. Prescribed fires caused smaller reductions in C and N storage (-46% and -35%) than wildfires (-67% and -69%). These differences are likely because of lower fuel loads or less extreme weather conditions in prescribed fires, both factors that result in lower fire intensity. Burned forest floors recovered their C and N pools in an average of 128 and 103 years, respectively. Among mineral soil layers, there were no significant changes in C or N storage, but C and N concentrations declined significantly (-11% and -12%, respectively). Mineral soil C and N concentrations were significantly reduced in response to wildfires, but not after prescribed burning.

Forest fires produce charcoal, or black carbon, some of which can be preserved over centuries and millennia in soils. Dissolved black carbon (DBC) from burning of the Brazilian Atlantic forest continued to be mobilized from the watershed each year in the rainy season, despite the fact that widespread forest burning ceased in 1973 (Dittmar *et al.*, 2012).

A large field study in the Amazon (225 forest plots) on the effects of anthropogenic forest disturbance (selective logging, fire, and fragmentation) on soil carbon pools showed that the first 30 cm of the soil pool did not differ between disturbed primary forests and undisturbed areas of forest, suggesting a resistance to impacts from selective logging and understory fires (Berenguer *et al.*, 2014). As with deforestation, impacts of human disturbances on the soil carbon are of particular concern in tropical forests located on organic soils and on steep easily-eroded slopes.

4.2 Shifting cultivation

Shifting cultivation practices, where land is cleared through fire, have been practiced for thousands of years, but recent increasing demographic pressure has reduced the duration of the fallow period, affecting the system sustainability. Moreover, especially in Southeast Asia where urbanisation is expanding in fertile planes, shifting cultivation is practiced in sloping uplands, which are prone to soil and carbon loss by erosion (Chaplot *et al.*, 2005). A review by Ribeiro Filho *et al.* (2013) reported negative impact on SOC associated with the conversion stage, modified by the characteristics of the burning. Chop-and-mulch of enriched fallows appears to be a promising alternative to slash-and-burn, conserving soil bulk density, and significantly increasing nutrient concentrations and organic matter content compared to burnt cropland, and a control forest in a study in the Amazon (Comtea *et al*., 2012).

4.3 Grassland management and dryland degradation

Grasslands, including rangelands, shrublands, pastureland, and cropland sown with pasture and fodder crops, cover 26% of the global ice-free land area and 70% of the agricultural area, and contain about 20% of the world's soil organic carbon (C) stocks. Grasslands on every continent have been degraded due to human activities, with about 7.5% of grassland having been degraded because of overgrazing (Conant, 2012). A meta-analysis (McSherry & Ritchie, 2013) of grazer effects on SOC density (17 studies that include grazed and un-grazed plots) found higher grazing intensity was associated with increased SOC in grasslands dominated by C4 grasses (increase of SOC by 6–7%), but with lower SOC in grasslands dominated by C3 grasses (decrease of SOC by an average 18%). An increase in mean annual precipitation of 600 mm resulted in a 24% *decrease* in the magnitude of the grazer effect on finer textured soils, but on sandy soils the same increase in precipitation produced a 22% *increase* in the grazer effect on SOC (McSherry & Ritchie, 2013).

Land use dynamics and climate change are the major drivers of dryland degradation with important feedbacks, with changes in plant community composition (e.g. shrub encroachment and decrease in vegetation cover; D'Odorico *et al*., 2013). A review by Ravi *et al*. (2010) indicated soil erosion as the most widespread form of land degradation in drylands, with wind and water erosion contributing to 87% of the degraded land. Grazing pressure, loss of vegetation cover, and the lack of adequate soil conservation practices increase the susceptibility of these soils to erosion. The degree of plant cover is negatively related to aridity, and an analysis of 224 dryland sites (Delgado-Baquerizo *et al*., 2013) highlighted a negative effect of aridity on the concentration of soil organic C and total N, but a positive effect on the concentration of inorganic P, possibly indicating the dominance of physical processes such as rock weathering, a major source of P to ecosystems, over biological processes that provide more C and N through litter decomposition (Delgado-Baquerizo *et al*., 2013).

Soil carbon dynamics in pastures strongly depend on management, with soil carbon increases or decreases observed for different combinations of animal densities and grazing frequency (Conant 2012; Machmuller *et al.* 2015). Different grazing strategies, especially in the semi-natural dryland biomes, have large implications for vegetation and the carbon balance (Yates *et al.* 2000). Under certain conditions, grazing can lead to increased annual net primary production over un-grazed areas, particularly with moderate grazing in areas with a long evolutionary history of grazing and low primary production but this does not always lead to an increase in soil carbon (e.g. Badini *et al.* 2007); grazing, like crop harvest, fundamentally leads to the rapid oxidation of carbon that would otherwise be eventually transferred to the soil. It has long been recognised that the potential effects of management on carbon storage in grassland and dryland soils are substantially greater than that of climate change or CO_2 enhancement (Ojima *et al.* 1993), and Henderson *et al.* (2015) estimated that the optimization of grazing pressure could sequester 148 Tg CO_2 yr^{-1}.

4.4 Artificial surfaces, urbanisation and soil sealing

In 2014, 54% of the world's population was urban, and by 2050, two thirds of the global population will be urban. Many regions in the world, (such as Europe and Asia) are affected by migration of populations from rural area to large megacities. Africa and Asia have more rural populations, but are urbanizing faster than the other regions (World Urbanization Prospects, 2014). With urbanization comes land take (development of scattered settlements in rural areas, the expansion of urban areas around an urban nucleus, and densification on land within an urban area) and soil sealing. Soil sealing refers to the permanent covering of an area of land and its soil by impermeable artificial material (e.g. asphalt and concrete), for example through buildings and roads. The area actually sealed is only part of a settlement area, and gardens, urban parks and other green spaces are not covered by an impervious surface (Prokop *et al*., 2011).

Sealing by its nature has a major effect on soil, diminishing many of its benefits (Tóth *et al*., 2007). It is normal practice to remove the upper layer of topsoil, which delivers most of the soil-related ecosystem services, and to develop a strong foundation in the subsoil and/or underlying rock to support the building or infrastructure. Loss of ecosystem and social services (mainly on high-quality soils) include impacts on water resources (e.g. reduction of rainfall absorbed by the soil, reduction of soil water holding capacity affecting flooding), on soil biodiversity when sealing prevents recycling of dead organic material (Marfenina *et al.* 2008), on the carbon cycle due to topsoil and vegetation removal (Davies *et al*., 2011). Sealing through expansion of urban areas can also lead to agricultural land becoming more marginal since the best agricultural land around settlements is lost as they expand, with agricultural land displaced to more marginal land.

Appropriate mitigation measures can be taken in order to maintain some of the soil functions. In urban planning management, objectives to reduce the impact of soil sealing include: i) preventing the conversion of green areas, ii) re-use of already built-up areas (e.g. brownfield sites Meuser, 2010; Hester & Harrison, 2001 – though remediation of contaminated sites can be costly; Maderova & Paton, 2013), iii) using (where appropriate) permeable cover materials instead of concrete or asphalt supporting green infrastructure, and iv) implementation of compensation measures. In order to deliver this mitigation a number of actions are necessary, e.g. reduction of subsidies that act as drivers for unsustainable land take and soil sealing (Prokop *et al*., 2011), and strong collaboration between relevant public authorities and governance entities (Siebielec *et al.*, 2010). Development impacts can be reduced by inclusion of green infrastructure, a network of high-quality green spaces and other environmental features that have a positive effect on well-being (Gill *et al.,* 2007) as well as soils. In some regions, urban sprawl is exacerbated by insufficient incentives to re-use brownfield (derelict, underused or abandoned former industrial or commercial) sites, putting increasing pressure on greenfield land take.

Actions to alleviate pressures on soils driven by sealing fall into three categories: limiting, mitigating and compensating. Actions to limit soil sealing centre around reduction of land take through development of spatial urban planning and environmental protection. Mitigation of soil sealing entails use of strategic environmental assessment for plans and programmes, use of permeable materials and surfaces, green infrastructure within built and urban environments, and natural water harvesting. Compensating soil sealing entails reclamation of degraded land, re-use of extracted topsoil, de-sealing and is incentivised by land take fees and application of environmental cost calculations.

5. Anthropogenic environmental change pressures that interact with land management pressures on soils

In addition to the direct impacts of humans on soils *via* land use change and land management, anthropogenic activity has indirect impacts through human-induced environmental change, such as pollution and climate change. These interact with land management. Soils provide a temporary but labile store for pollutants (Meuser, 2010). Natural processes can release pollutants back to the atmosphere, make them available to be taken up by plants and organisms, leached in to surface waters (Galloway *et al.*, 2008) and/or transported to other areas by soil erosion (Ravi *et al.*, 2010). Pollutants disrupt natural biogeochemical cycles by altering both soil quality and function through direct changes to the nutrient status, acidity and bioavailability of toxic substances and also by indirect changes to soil biodiversity, plant uptake and litter inputs (EEA, 2014). Soil sensitivity to atmospheric pollution varies with respect to key properties influenced by geology (cation exchange capacity, soil base saturation, aluminium), organic matter, carbon to nitrogen ratio (C:N) and water table elevation (EEA, 2014).

Atmospheric pollutant deposition impacts on soils vary with respect to soil sensitivity to a specific pollutant and the actual pollutant load. Sulphur, nitrogen and heavy metals are released in to the atmosphere by fossil fuel combustion (e.g. power generation, industry and transport) and non-combustion processes (e.g. agricultural fertilizers, waste). These pollutants are transported off-site and deposited as either dry or wet deposition, which can cross national borders. Deposition is enhanced in forests and with altitude because of reduced wind speeds and greater precipitation, respectively, impacting remote areas. Harmful effects to soil function and structure occur where deposition exceeds the 'critical load' that a particular soil can buffer (Nilsson & Grennfelt, 1988). Spatial differences in soil sensitivity (commonly defined by the 'crucial load') and pollutant deposition result in an uneven global distribution of impacted soils (Figure 3). For instance, global emissions of sulphur and nitrogen have increased 3-10 fold since the pre-industrial period (van Aardenne *et al.*, 2001), yet only 7-17% of the global land area sensitive to acidification is in a region where deposition exceeds the critical load (Bouwman *et al.*, 2002).

Emissions of pollutants, notably sulphur, across Europe and North America have declined since the 1980s following protocols established under the 1979 Convention on Long-range Transboundary Air Pollution (LRTAP) and the 1990 US Clean Air Act Amendments (CAAA) (Greaver *et al.*, 2012; Reis *et al.*, 2012; EEA, 2014). Conversely, emissions are likely to increase in response to industrial and agricultural development in south and east Asia, sub-Saharan Africa and South America (Kuylenstierna *et al.*, 2001; Dentener *et al.*, 2006). Further emission increases are occurring in remote areas due to mining activity, such as oil sand extraction in Canada (Kelly *et al.*, 2010; Whitfield *et al.*, 2010).

5.1 Sulphur deposition

Sulphur emissions are primarily from combustion of coal and oil, typically associated with power generation and heavy industry. In 2001, regions with deposition in excess of 20 kg S ha^{-1} yr^{-1} where China and Republic of Korea, western Europe and eastern North America (Vet *et al*., 2014; Figure 3a). Deposition in un-impacted areas is <1 kg S ha^{-1} yr^{-1} (Figure 3a). Pollution control measures have seen an 80% reduction in pollutant sulphur deposition across Europe between 1990 and 2010 (Reis *et al.*, 2012), and emissions in China have declined since 2005 (Fang *et al.*, 2013).

Soil acidification is a natural process that is altered and accelerated by sulphur and nitrogen deposition (Greaver *et al.*, 2012). Sulphur oxides (SO_x) react with water to form sulphuric acid (H_2SO_4). Excess inputs of acidity (H^+) displace soil base cations (e.g. calcium (Ca^{2+}) and magnesium (Mg^{2+})) from soil surfaces into solution, which are subsequently lost by leaching (Reuss & Johnson, 1986). Mineral soils can buffer base cation losses if inputs from rock weathering and/or atmospheric dust deposition exceed the amount lost. Therefore, the global distribution of acid sensitive soils is associated with conditions that favour development of soils with low cation exchange capacity and base saturation (Bouwman *et al.*, 2002; Figure 3c). Wetland can also buffer inputs of acidity through biological sulphate reduction, although acidity can be mobilised again following drought and drainage (Tipping *et al.*, 2003; Laudon *et al.*, 2004; Daniels *et al.*, 2008). Organic acids can also buffer mineral acidity in naturally acidic organic soils (Krug and Frink, 1983).

Decreased soil fertility or ‘sterilisation’ due to loss of nutrients and mobilisation of toxic metals, particularly Al, are caused by acidification. Impacts in Scandinavia over the 1960s-80s included declines in freshwater fish populations and damage to forests (EEA, 2014). Sulphur can also stimulate microbial processes that make mercury bioavailable, leading to bioaccumulation in the food chain (Greaver *et al.*, 2012). In agricultural soils in Europe, however, fertilizer inputs of sulphur have increased to combat crop sulphur deficiencies as a result of sulphur emission controls (Bender & Weigel, 2011).

Acidification is reversible, evident by increases in soil pH following decreased sulphur emissions, although the recovery time varies; some areas with organic soils where deposition has declined are showing either slow or no recovery (Greaver *et al.*, 2012; Lawrence *et al.*, 2012; RoTAP, 2012). On agricultural soils, lime can be applied to increase soil pH. However, 50-80% of sulphur deposition on land is on natural, non-agricultural land (Dentener *et al.*, 2006). Application of lime to naturally acidic forest soils can cause further acidification of deep soil layers whilst increasing decomposition in surface litter, with no improvement in tree growth (Lundström *et al.*, 2003).

Wider effects of acidification are starting to be understood through long-term monitoring. Decreased organic matter decomposition due to acidification has increased soil carbon storage in tropical forests (Lu *et al.*, 2014). However, in temperate forest soils acidification can lead to reduced C:N ratios in soil, which in turn increases nitrification (Aber *et al*., 2003), but on already acidic soils reduces nitrification. In wetland soils, methane (CH_4) emissions have also been suppressed by sulphur deposition (Gauci *et al.*, 2004). Conversely, declining sulphur deposition has been associated with increased dissolved organic carbon fluxes from organic soils (Monteith *et al*., 2007), and decreased soil carbon stocks in temperate forest soils (Oulehle *et al.*, 2011; Lawrence *et al.*, 2012).

5.2 Nitrogen deposition

Nitrogen deposition covers a wider geographical area than sulphur, as the sources are more varied, including extensive agriculture fertilizer application, ammonia derived from livestock operations, and biomass burning in addition to fossil fuel combustion (Figure 3b). Regions with deposition in excess of 20 kg N ha^{-1} yr^{-1} in 2001 were western Europe, South Asia (Pakistan, India, Bangladesh) and eastern China (Vet *et al.*, 2014); although extensive areas with 4 kg N ha^{-1} yr^{-1} were found across North, Central and South America, Europe and Sub-Saharan Africa. By contrast, 'natural' deposition in un-impacted areas is around 0.5 kg N ha^{-1} yr^{-1} (Dentener *et al.*, 2006). While emissions related to fossil fuel combustion have declined along with sulphur across Europe, agricultural sources of nitrogen are likely to stay constant in the near future across Europe (EEA, 2014), whilst overall global emissions are likely to increase (Galloway *et al.*, 2008). Nitrogen deposition in China's industrialized and intensively managed agricultural areas in the 2000s was similar to peaks in Western Europe during the 1980s before mitigation (Liu *et al.*, 2013).

Deposition of nitrogen induces a 'cascade' of environmental problems, including both acidification and eutrophication that can have both positive and negative effects on ecosystem services (Galloway *et al.*, 2003). Excluding agricultural areas where nitrogen is beneficial, 11% of land surface received nitrogen deposition above 10 kg N ha^{-1} yr^{-1} (Dentener *et al.*, 2006; Bouwman *et al*. 2002; Figure 3d). In Europe, eutrophication has and will continue to impact a larger area than acidification (EEA, 2014).

Nitrogen fertilisation can increase tree growth (Magnani *et al.*, 2007) and cause changes in plant species and diversity (Bobbink *et al.*, 2010), which in turn will alter the amount and quality of litter inputs in to soils, notably the C:N ratio and soil-root interactions (RoTAP, 2012). However, increased carbon sequestration (Reay *et al.*, 2008) may be offset by increased emissions of the greenhouse gases N_2O and CH_4 (Liu & Greaver, 2009). Long-term changes caused by nitrogen deposition are uncertain as transport times vary between environmental systems; and the only way to remove excess nitrogen is to convert it to an unreactive gas (Galloway *et al.*, 2008).

[Figure 3 here]

5.3 Heavy metal deposition

Heavy metal emissions are associated with coal combustion and heavy industry. In the UK, deposition is responsible for 25-85% of inputs to UK soils (Nicholson *et al.*, 2003). In Europe, the areas at risk from cadmium, mercury and lead deposition in 2000 were 0.34%, 77% and 42% respectively, although emissions are declining (Hettelingh *et al.*, 2006). Tighter legislation to control industrial emissions of heavy metals are helping to reduce the environmental load of heavy metals in many regions, though rapid industrial growth in some regions such as East Asia is increasing pressures on soil from heavy metal deposition. Global heavy metal emissions and deposition are poorly understood in comparison to sulphur and nitrogen; although the on-site impact of heavy metal contamination has been well studied (Guo *et al.*, 2014). Metals in bioavailable form have toxic effects on soil organisms and plants, influencing the quality and quantity of plant inputs to soils, rate of decomposition and, importantly, can bio-accumulate in the food chain. Some heavy metals will persist for centuries as they are strongly bound to soil organic matter (RoTAP, 2012), although they can be mobilised to bioavailable form following drought-induced acidification, drainage and soil erosion (Tipping *et al.*, 2003; Rothwell *et al.*, 2005).

Whilst the direct impacts of sulphur, nitrogen and heavy metals on inorganic soil chemical processes are generally well understood, many uncertainties still exist about pollutant impacts on biogeochemical cycling, particularly interactions between organic matter, plants and organisms in natural and semi-natural systems (Greaver *et al.*, 2012). Process understanding is dominated by research in Europe and North America (e.g. Bobbink *et al.*, 2010). Research is needed across Asia, Africa and South and Central America where soil properties and environmental conditions differ. Models need to be developed to examine the combined effects of air pollutants and their interactions with climate change impacts and feedbacks on greenhouse gas balances and carbon storage (Spranger *et al.*, 2008; RoTAP, 2012). Air quality, biodiversity and climate change polices all impact on soils. A more holistic approach to protecting the environment is needed, particularly as some climate change policies (e.g. biomass burning, carbon capture and storage) have potential to impact air quality and, therefore, soil quality (Reis *et al.*, 2012; RoTAP, 2012; Aherne & Posch, 2013).

Indirect impacts on soils can be addressed largely by preventing the pollution at source, or by mitigating the adverse effects where these have already occurred. Air pollution control on coal burning and increased car and fleet efficiency standards has been effective in reducing sulphur deposition in many areas of the world, particularly in Europe since the 1970s. Substitution of coal with bioenergy might also reduce sulphur emissions, but unless burned cleanly in a controlled way, can also release pollutants to the air. In terms of nitrogen, ammonia abatement techniques when fertilizers are spread (e.g. slurry injection) are helping to reduce N deposition (Sutton *et al.*, 2007).

6. Existing policies and practices that alleviate global change pressures on soils from land use and management

The previous text has highlighted specific anthropogenic activities that exert or alleviate pressures on soils. Actions that alleviate pressures on soils driven by land use change and land management can be broadly divided into three categories, those that:

1) Prevent conversion of natural ecosystems to other uses (e.g. protected areas, reduced deforestation, prevention of wetland drainage, intensification rather than extensification);
2) Prevent soil degradation (erosion control, fire management, reduced tillage / conservation agriculture, long term fallows, flood protection, use of organic amendments, intercropping, improved rotations); and
3) Result in soil / ecosystem restoration (e.g. peatland rewetting, afforestation, re-vegetation on degraded lands, improved grass varieties, appropriate animal stocking densities, bioremediation).

Policies to encourage such actions were recently reviewed by Bustamante *et al*. (2014) and include:

a) Economic incentives, e.g., special credit lines for low carbon agriculture and forestry practices and projects, payment for ecosystem services (such as carbon storage) and tradable credits such as carbon,
b) Regulatory approaches, e.g. enforcement of environmental law to protect natural areas, set-aside policies,
c) Research, development and diffusion investments, e.g. increase of resource use-efficiency, livestock improvement,
d) Information and certification schemes, e.g. in China, forest certification to promote sustainable forest management, state regulation for protecting mandatory arable lands, protection projects on Tibetan grasslands, a national wetland protection programme, and the "grain for green" programme.

Many of these actions and policies are not directed at soil conservation, but nevertheless have an effect on soil quality. Two of the main pieces of international policy that have served to reduce pressures on soils, directly and indirectly, are the United Nations Convention to Combat Desertification (CCD) and the United Nations Framework Convention on Climate Change (UNFCCC). In general, policies and actions are important at all scales from international conventions to local action, and local activity is encouraged by policies at regional, national and global level. Policies to sustainably increase land productivity, for example, can prevent land use change, and there are various other supporting actions that can help deliver improvements, e.g. agricultural research, technology transfer, knowledge transfer and improved rural infrastructure. Some examples of policies that impact on land management and soil quality are given below.

6.1 United Nations Framework Convention on Climate Change (UNFCCC) and other climate specific policies

Soil carbon storage and nutrient cycling as climate services are being increasingly recognised e.g. under UNFCCC as part of national reporting and accounting, as part of life-cycle greenhouse gas assessments for biofuels, in various regional initiatives and national efforts. The UNFCCC is an international treaty, which came into force in 1994, setting an overall framework for intergovernmental efforts to tackle the challenge posed by climate change. The requirements for the 196 country Signatories (or 'Parties') to the UNFCCC include adopting national mitigation policies and publishing national inventories of anthropogenic emissions and sinks of greenhouse gases including activities on the land such as afforestation, deforestation, agricultural management and wetland drainage and rewetting. Developed country signatories have legally binding targets under the Kyoto Protocol and can count land based emissions or sinks towards meeting these targets, thus incentivising activities that protect soil carbon. Developing countries currently have voluntary targets and several countries have made pledges that include reduced deforestation (e.g. Brazil and Indonesia) or afforestation (e.g. 400000 km^2 in China). Under the Clean Development Mechanism (CDM) developed countries can fund projects in developing countries that generate certified emission reduction credits (CERCs). China, for example, has the largest number of CERCs in the world (IFPRI, 2011). Brazil also has 180 CDM projects, the third largest number of CERCs after China and India (Cole & Liverman, 2011). A number of projects in Africa, North America and South Asia have a significant component for soil greenhouse gas emission reduction of soil carbon sequestration, financed through the Verified Carbon Standard or the American Carbon Registry.

As part of negotiations leading to the new climate treaty in Paris in December 2015, all parties will be required to submit INDCs (Intended Nationally determined Contributions). The new treaty will also include provision for REDD+ (reduced Emissions from Deforestation and Degradation, including management of forests and enhancement of forest carbon stocks). This could go some way to protecting forest soils, and negotiations have been intense around methods for monitoring reporting and verification, with key issues such as permanence (the risk the forest may be lost at a later date due to management or environmental change) and leakage (displacement of land use change to other areas), and how to finance such activities.

6.2 United Nations Convention to Combat Desertification (CCD)

The CCD entered into force in December 1996; today 179 countries acknowledge it as a legally binding framework to tackle land degradation and promote sustainable development in fragile ecosystems. The Global Mechanism was established under the convention to "promote actions leading to the mobilization and channelling of substantial financial resources, including for the transfer of technology, on a grant basis, and/or on concessional or other terms, to affected developing country Parties". In September 2011 the United Nations General Assembly declared a goal of building a world with no land degradation. In October 2011 parties to the CCD issued a declaration calling for zero land degradation and for adopting sustainable land management as a way to achieve sustainable development.

6.3 Millennium Development Goals (MDGs)

Of the eight MDGs (UNDP, 2014a), soil protection is most relevant to the goal to ensure environmental sustainability, since soils are critical in underpinning environmental sustainability (Smith *et al.*, 2015). A complementary MDG, to develop a global partnership for development, will improve the governance structure to deliver soil security. The other MDG to which soils plays a critical contribution is the goal to eradicate extreme poverty and hunger, with the role of soils in supporting food provision critical for the latter part of this MDG (Smith et al., 2015). The MDGs are currently being revisited to set a post-2015 development agenda (UNDP, 2014b), with discussion around the themes of localising the post-2015 development agenda, helping to strengthen capacities and build effective institutions, participatory monitoring for accountability, partnerships with civil society, engaging with the private sector, and culture and development. The key emerging principles from the dialogue are participation, inclusion, and the need for strengthened capacities and partnerships (UNDP, 2014b). It is important that soils play their role in delivering this post-2015 agenda.

6.4 Protected areas and the Convention on Biological Diversity (CBD)

Many measures to protect biodiversity and vulnerable habitats also protect the soils underpinning them, so numerous conservation actions around the world serve to protect soils, even if this was not the primary aim (Smith *et al*., 2013). Between 1990 and 2010, the amount of forest land designated primarily for the conservation of biological diversity increased by 35 percent, indicating a political commitment to conserve forests. These forests now account for 12 percent of the world's forests (FAO, 2010). In India, a Supreme Court ruling in 2011 on effective self-governance of "common" or communal land by local communities may help to protect these valuable resources, and thereby the soils that underpin them. Soil biodiversity is known to be important for soil function (Bodelier, 2011), yet it rarely receives the attention enjoyed by larger flora and fauna within the ecosystem.

6.5 Reduced deforestation and forest management

Various actions have been implemented to reduce deforestation (Bustamante *et al*., 2014), and to reduce the impact of forestry activities, such as reduced impact logging. UNFCCC, carbon markets and other international environmental programs have contributed to global efforts to reduce deforestation in addition to other sustainable natural resource management programs in countries and by industry. For example, zero deforestation commitments made by several companies (many made in 2014), and activities from bodies such as the Roundtable for Sustainable Palm Oil (RSPO) and the Forest Stewardship Council (FCO) certification scheme. Land improvement has increased in East Asia between 1981 and 2006 despite population increase, attributed largely to policies promoting tree planting and forest plantation programs in China and Korea. In Brazil, deforestation was rapidly reduced after national laws and regulations were enacted to protect forests in the 1990s and early 2000s (including the soy moratorium and the forest code), followed up by state and municipal governments setting further by-laws enforcing the deforestation moratorium (Bustamante *et al.*, 2014).

6.6 Agricultural policies and practices

The pressures on soils imposed by land use intensity change can be mitigated by regulation of over-grazing and reduction of over-stocking on grazed grasslands, return of crop residues to the soil, reduced tillage, best management practices, targeted nutrient management and precision farming on croplands, and wetland / floodplain restoration. These actions have been encouraged by various policies. Some examples include: The EU set-aside programme of the 1990s encouraged less intensive use of agricultural land where production is low and environmental impacts are high. The EU Common Agricultural Policy ties agricultural subsidies to implementation of best management practices and environmental protection, for example through pillar 2 (rural development programmes) providing crop insurance for lower fertilizer application rates; in Africa, policies for integrated land management to help protect vulnerable soils; China's conservation tillage program (2012-2030); the USA Conservation Reserve Program (set aside marginal lands, steep slopes).

7. Conclusion: Keeping soils central to the science and policy agendas

The International Year of Soils in 2015 is an excellent opportunity to raise the profile of soils in the minds of national and international policy makers, land managers, timber and agro-industries, and the public. Ensuring that vulnerable and high environmental value soils (e.g. peatlands) are considered when making policies and decisions about which habitats and ecosystems to convert or to protect, will help to reduce the pressure on soils particularly vulnerable to global change drivers such as land use and land management, and maintain important ecosystem services. This is in part happening with agendas around valuation of ecosystem services and life-cycle assessments of impacts of land use change that include soil carbon. At a time when governments are negotiating a legally binding climate change treaty and making national targets for greenhouse gas reduction, and revisiting the Millennium Development Goals, keeping soil carbon and nitrogen central to land based greenhouse gas monitoring and reporting will maintain awareness with policy makers and industries with emissions reduction targets. Both science and policy agendas are increasingly concerned with long-term food security, ensuring that soils are central to considerations of how to achieve on-going increases in production will enable those increases to be more sustainable into the future.

Research and policy regarding soil quality and sustainability is abundant, but patchy and disjointed. To ensure that soils are protected as part of on-going wider environmental and sustainable production efforts, soils cannot, and should not, be considered in isolation of the ecosystems that they underpin, but the role of soils in supporting ecosystems and natural capital needs greater recognition (Robinson *et al*., 2013, 2014). This can, in part, be enhanced through education and awareness-raising which has started during the International Year of the Soils in 2015. The time is ripe to consider a global soil resilience programme, under the auspices of a global body such as the UN or one of its delivery agencies such as the FAO to monitor, recover or sustain soil fertility and function, and to enhance the ecosystem services provided by soils. The lasting legacy of the International Year of Soils in 2015 should be to bring together robust scientific knowledge on the role of soils, and to put soils at the centre of policy supporting environmental protection and sustainable development.

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Tables

Table 1. Observed and modelled soil carbon change (%) when converting from land cover classes in the left hand column to land cover classes listed across the top. Results are from meta-analysis of observations from the sources listed below. Model results (range across three models) are shown for comparison in square brackets, range across the ISAM, LPJml, and LPJ_GUESS models (see text), although note this calculated as difference in soil carbon under the different land classes in 2010 and is thus not modelled loss/gain after a conversion. Negative numbers represent loss of soil carbon.

		Regrowth Forest	Tree plantation	Grassland	Pasture	Cropland
Forest	Global		-13% (3)[a]		+8% (3)	-42% (3)
	Trop.	-9% (2)			-12% (2)	-41% (1)
						-25% (2)[b]
						-30% (2)[c]
						-24% (5)
					[-40 to -63%]	[-51 to -62%]
	Temp.					-52% (1)
						-36% (4)
					[-52% to +17]	[-24 to -60%]
	Boreal					-31% (1)
					[-14 to -49%]	[-63 to -65%]
Grassland	Global					
	Trop				[-1 to +15%]	[-2 to -6%]
	Temp					-32% (4)
					[-28 to +3%]	[-15 to -53%]
	Boreal				[-26 to -71%]	[-70 to -79%]
Pasture	Global		-10% (3)			-59% (3)
	Trop					[-19 to +0.5%]
	Temp					[-17 to -35%]
	Boreal					[-28 to -59%]
Cropland	Global	+53% (3)	+18% (3)		+19% (3)	
	Trop		+29% (2)		+26% (2)	
	Temp	+16% (4)	+20% (6)	+28% (4)		
	Boreal					

Footnotes: [a] Broadleaf tree plantations onto prior native forest or pasture did not affect soil C stocks whereas pine plantations reduced soil C stocks by -12 to -15%; [b] Annual crops; [c] Perennial crops; 1 Wei *et al.* (2014a); 2 Don *et al.* (2011); 3 Guo & Gifford (2002; tropical and temperate zones compiled); 4 Poeplau *et al.* (2011); 5 Murty *et al.* (2014); 6 Barcena *et al.* (2014).

Table 2. Soil carbon loss due to land use change 1860 to 2010 ($PgCO_2$)

Model	Tropical	Temperate	Boreal	Global
LPJ-GUESS	46	55	1	109
LPJmL	128	95	0	227
ISAM	63	139	19	221
Mean	79	96	7	186

Table 3. Threats to soil resource quality and functioning under increasing intensity of agricultural management

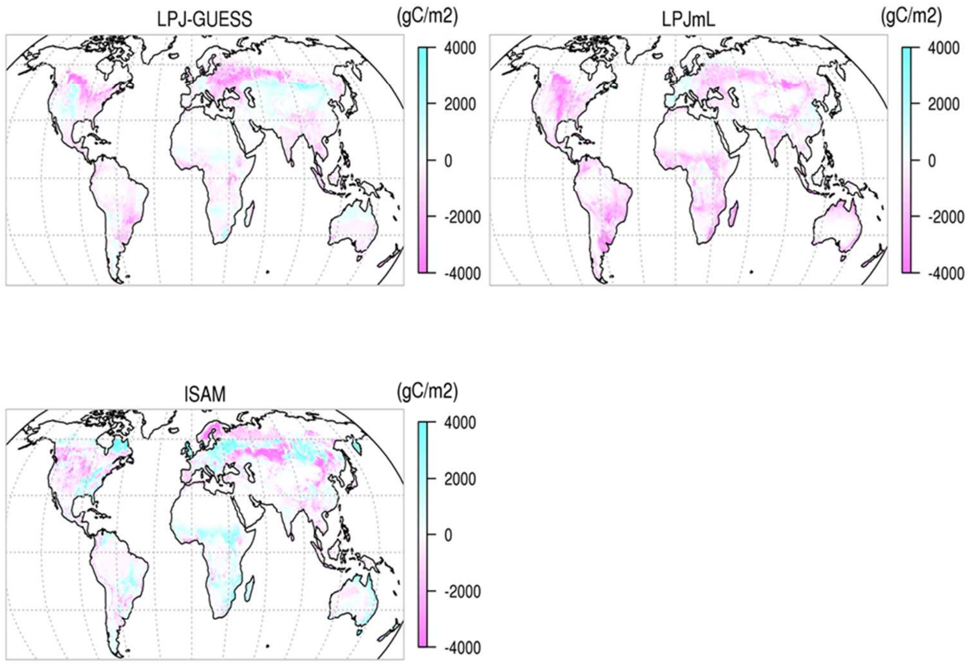
Agricultural management practice	**Specific issue**	**Distribution**	**Major environmental consequence**	**Knowledge gap**
Cropping practice	Harvest frequency	Global	Soil quality and resilience	Impact on total C and nutrient cycles
	Monoculture	Global but particularly in developing and transition countries	Soil health, pesticide residue in intensively managed monocultures	Biological resilience
Use of agrochemicals	Over fertilization	Particularly in some developing countries	Soil acidification, water pollution, N_2O emission and nitrate accumulation	Rate reducing versus balancing
Irrigation	Submerged Rice	Developing countries, Asian	Water scarcity, methane emission	Trade-offs C and water,
	Arid/semi-arid regions	Arid/semi-arid regions	Secondary salinization, water scarcity	Competition use of water
Livestock management	Over-grazing	Largely in developing countries	Soil degradation, water storage, C loss	Forage versus feed crops?
	Industrial breeding	Largely in industrialized and transition countries	Waste pressure, water pollution, residue of veterinary medicine and antibiotics	Safe waste treatment and recycling
Agriculture in wetlands	Wetland drainage	Developing and transition countries	C loss	Agro-benefit versus natural value

Figure Legends

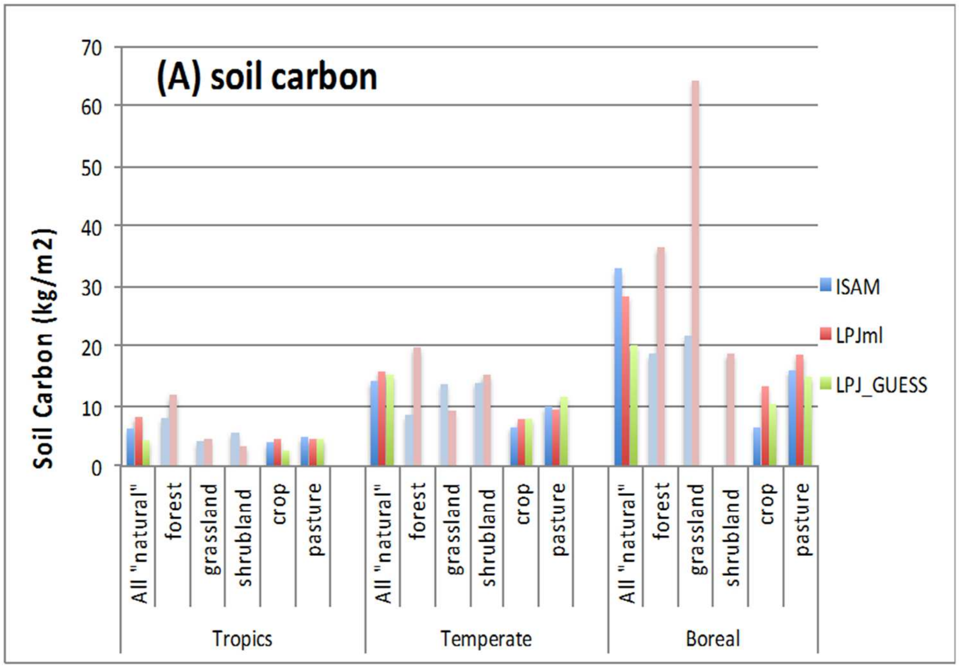
Figure 1. Maps of change in soil carbon due to land use change land and land management from 1860 to 2010 from three vegetation models. Pink indicates loss of soil carbon, blue indicates carbon gain.

Figure 2. Soil carbon and nitrogen under different land cover types in three different vegetation models (values are the annual average over the period 2001 to 2010).

Figure 3. Uneven global distribution of soils sensitive to pollution by (a) acidification and (b) eutrophication (measured by soil C:N) compared to uneven distribution of atmospheric (c) sulphur and (d) nitrogen pollution. Soils most sensitive to acidification have low base saturation and cation exchange capacity, as defined by (Kuylenstierna *et al.*, 2001). Acidification is caused by both sulphur and nitrogen. Eutrophication is caused by nitrogen. Soil data in (a) and (b) were produced using the ISRIC-WISE derived soil properties (ver 1.2) (Batjes, 2012) and the FAO Digital Soil Map of the World. Atmospheric deposition data in (c) and (d) were provided by the World Data Centre for Precipitation Chemistry (http://wdcpc.org, 2014) and are also available in Vet *et al.* (2014). Data show the ensemble-mean values from the 21 global chemical transport models used by the Task Force on Hemispheric Transport of Air Pollution (HTAP) (Dentener *et al.*, 2006). Total wet and dry deposition values are presented for sulphur, oxidized and reduced nitrogen.



254x190mm (96 x 96 DPI)



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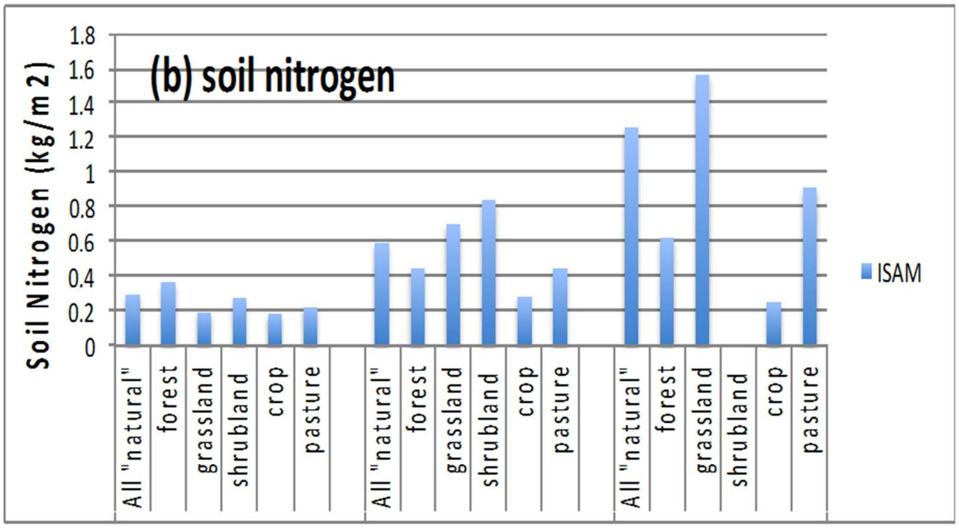

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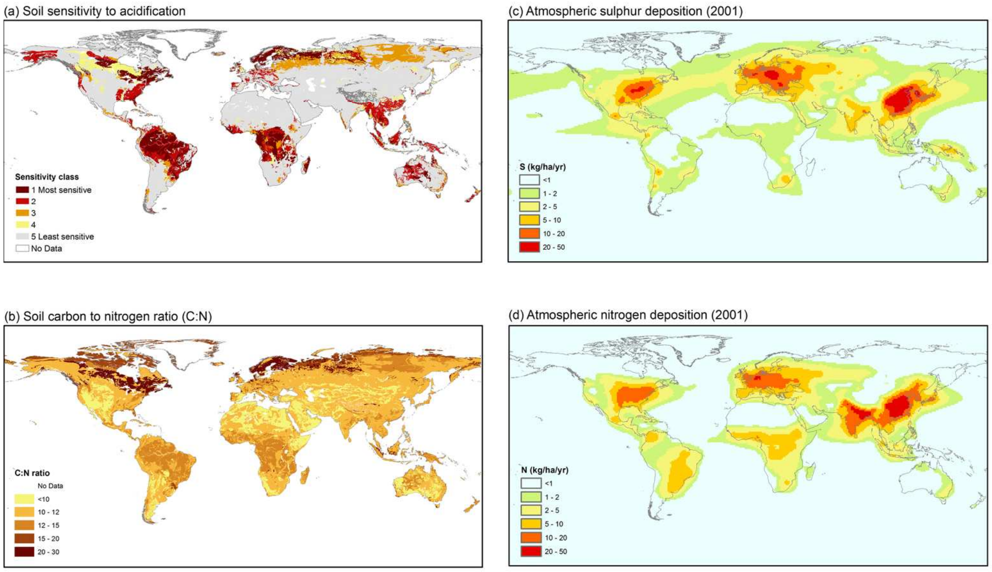

168x101mm (300 x 300 DPI)